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LAND USE IN LCA

A spatially explicit data-driven approach to assess the effect of agricultural land occupation on species groups

Pieter M. F. Elshout · Rosalie van Zelm · Ramkumar Karuppiah · Ian J. Laurenzi · Mark A. J. Huijbregts

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Abstract
Purpose Change of vegetation cover and increased land use intensity, particularly for agricultural use, can affect species richness. Within life cycle impact assessment, methods to assess impacts of land use on a global scale are still in need of development. In this work, we present a spatially explicit data-driven approach to characterize the effect of agricultural land occupation on different species groups.
Methods We derived characterization factors for the direct impact of agricultural land occupation on relative species richness. Our method identifies potential differences in impacts for cultivation of different crop types, on different species groups, and in different world regions. Using empirical species richness data gathered via an extensive literature search, characterization factors were calculated for four crop groups (oil palm, low crops, Pooideae, and Panicoideae), four species groups (arthropods, birds, mammals, and vascular plants), and six biomes.
Results and discussion Analysis of the collected data showed that vascular plant richness is more sensitive than the species richness of arthropods to agricultural land occupation. Regarding the differences between world regions, the impact of agricultural land use was lower in boreal forests/taiga than in temperate and tropical regions. The impact of oil palm plantations was found to be larger than that of Pooideae croplands, although we cannot rule out that this difference is influenced by the spatial difference between the oil palm- and Pooideae-growing regions as well. Analysis of a subset of data showed that the impact of conventional farming was larger than the impact of low-input farming.
Conclusions The impact of land occupation on relative species richness depends on the taxonomic groups considered, the climatic region, and farm management. The influence of crop type, however, was found to be of less importance.

Keywords Biodiversity · Characterization factor · Crop cultivation · Life cycle impact assessment · Land occupation · Species richness

1 Introduction

Agricultural land use can directly affect species diversity in a region (Foley et al. 2005; Matson et al. 1997; Vitousek et al. 1997) and has been deemed to be the most influential driver for biodiversity loss to date (Millennium Ecosystem Assessment 2005). The impact of agricultural land use on species diversity worldwide depends on several factors. Firstly, crops are grown under different climatic conditions with different ways of cultivation, which can lead to a large geographical variability in the environmental impact of crop cultivation (Holland 2004; McLaughlin and Mineau 1995). Moreover, growing a specific crop in a tropical region may require clearance of rainforest, while the same crop may replace natural grasslands in temperate regions. Species from these different regions may react differently to agricultural land use as the structure of the cropland vegetation will resemble the natural vegetation structure to a greater or lesser extent. This makes the impact of land occupation region-

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P. M. F. Elshout (✉) · R. van Zelm · M. A. J. Huijbregts
Institute for Water and Wetland Research, Department of Environmental Science, Radboud University Nijmegen, P.O. Box 9010, 6500 GL Nijmegen, The Netherlands
e-mail: p.elshout@science.ru.nl

R. Karuppiah · I. J. Laurenzi
ExxonMobil Research and Engineering Company, 1545 Route 22 East, Annandale, NJ 08801-3059, USA

specific. Most research on the impact of agricultural land use has been done in tropical regions where crops like oil palm, rubber, and soybean replace the natural rainforest and potentially impoverish the faunal community (Danielsen et al. 2008; Gardner et al. 2009; Nepstad et al. 1999; Sodhi et al. 2004; Wright and Muller-Landau 2006).

Secondly, some types of cropland may be more suitable as habitat to plants and animals than others, not only because of differences in farming practices but also due to differences in crop structure. Vegetation density influences the success of concealing from a predator or prey, provides shelter against extreme weather conditions, and influences the diversity and accessibility of food items (Cody 1981; Wilson et al. 2005). A few studies have assessed the impact of different crops on biodiversity. Booij and Noorlander (1992) found small differences in ground beetle species densities in six row crop systems, showing that species densities were largest in wheat, sugar beet, and pea fields. Perennial crops such as switchgrass and miscanthus provide greater habitat stability than annual row crops and were found to support a larger abundance and diversity of insects than corn (Andow 1991; Gardiner et al. 2010; Ward and Ward 2001), as well as a larger diversity of birds than wheat (Bellamy et al. 2009).

Thirdly, different taxonomic groups may respond in a different way to agricultural land use change and intensification due to variations in, for example, size, mobility, and diet. Also, farmers may actively control species (groups) that are considered pests to a specific crop (Flohre et al. 2011; Kessler et al. 2009). Schulze et al. (2004) found that the decline of bird and plant species when changing a primary forest into agroforestry and cropland is larger than the decline of butterflies and dung beetles. Kessler et al. (2009) found that replacement of a mature forest with agroforestry caused species richness of trees, dung beetles, and birds to decline, of ants to stay equal, and of herbs and canopy beetles to increase. A meta-analysis by Danielsen et al. (2008) showed that species richness of vertebrates was always lower in oil palm plantations than in tropical forests while no difference was found for invertebrates.

There has been an increased effort to include the impacts of land use in life cycle assessment (LCA) during the last few decades (e.g., Bare 2011; Brentrup et al. 2002; Kløverpris et al. 2007; Köllner and Scholz 2007; Milà i Canals et al. 2007; Müller-Wenk and Brandão 2010; De Baan et al. 2013a). Multiple land quality indicators have been proposed to quantify land use impacts, including ecological soil quality, biotic production potential, and biodiversity (Milà i Canals et al. 2007). In the case of biodiversity, the impact has often been quantified by deriving characterization factors (CFs) based on the relative difference between the species composition (e.g., species richness, abundance, evenness, and/or naturalness) during land use and that in a (semi)natural reference situation (e.g., De Baan et al. 2013b; Curran et al. 2011; Köllner and Scholz 2008; Schmidt 2008; De Schryver et al. 2010; De Souza et al. 2013; Vogtländer et al. 2004; Weidema and Lindeijer 2001). Impact assessment methods are highly dependent on data availability, and many LCA studies are biased toward well-studied taxonomic groups and geographic regions: Taxonomic coverage is often limited to vascular plants (e.g., Lindeijer 2000a; Schmidt 2008; De Schryver et al. 2010; Vogtländer et al. 2004), and most studies focus on northern Europe (Köllner 2000; Köllner and Scholz 2008; Michelsen 2008; De Schryver et al. 2010; Vogtländer et al. 2004), North America (Geyer et al. 2010), and Southeast Asia (Schmidt 2008). Moreover, different crop types are often considered as one group when croplands are compared to other land cover types (e.g., "arable land"; Köllner and Scholz 2008; Schmidt 2008; De Schryver et al. 2010), or they are grouped into, e.g., annual versus permanent crops (De Baan et al. 2013b; De Souza et al. 2013) or grain crops versus row/field crops (Geyer et al. 2010).

Within LCA, three phases of land use are distinguished: (1) land transformation, i.e., the impact of making the land suitable for a new activity; (2) land occupation, i.e., the impact during the new activity; and (3) land relaxation, i.e., recovery of the land after the activity has ended (Köllner and Scholz 2007; Lindeijer et al. 2002; Milà i Canals et al. 2007). The goal of the present study was to increase understanding of the factors that influence the impact of land occupation on species richness, identify data gaps, and facilitate the development of more detailed CFs for land occupation in the future. We aimed to identify potential differences in (1) the impact of cultivation of different crop types, (2) the impact of crop cultivation in different world regions, and (3) the sensitivity of different species groups to crop cultivation. Crop-, species-, and biome-specific CFs were calculated based on data gathered via an extensive literature search, using a method recently described by De Baan et al. (2013b). Our method builds upon the method of De Baan et al. (2013b) by expanding the available data set, differentiating between crop types, and taking into account farm management strategies in the analysis.

2 Methods

2.1 Framework

Characterization factors (CFs) for the impact of agricultural land occupation were determined following the linear

relationship described by Köllner and Scholz (2008) and De Baan et al. (2013b):

$$CF_{x,i,y} = 1 - \frac{S_{crop,x,i,y}}{S_{ref,i,y}} \quad (1)$$

where S_{crop} and S_{ref} are the observed species richness (number of species) of species group y when cultivating crop type x in region i and the observed species richness of the reference land cover in region i, respectively. In this study, we use natural ecosystems as a reference, which allows the species richness of the cropland to be compared with the number of species that may have existed in that location if the land occupation would not have taken place. The equation yields characterization factors between $-\infty$ and $+1$, where a negative value means a positive effect of land occupation (i.e., a larger species richness) and the maximum of 1 represents a 100 % decline in species richness. A low CF indicates lower impact of land occupation on biodiversity.

2.2 Data collection and processing

Data were collected for corn, sugarcane, sugar beet, soybean, oil palm, rapeseed, switchgrass, sorghum, sunflower, jatropha, cassava, potato, wheat, barley, and rye, which include major second-generation biofuel feedstocks. For the CF calculations, publications on species richness on agricultural land (up to April 2012) were searched within the ISI Web of Knowledge database using the following search key:

TS = ((biodiversit* OR ″species richness*″ OR ″species abundanc*″) AND (corn* OR maize* OR ″sugarcane*″ OR ″sugar beet*″ OR soy* OR ″oil palm*″ OR rape* OR switchgrass* OR sorghum* OR sunflower* OR jatropha* OR cassava* OR potato* OR wheat* OR cereal* OR rye* OR barley*))

This resulted in a total of 2,591 hits. The summaries of these publications were checked in order to select those studies that provide data on the occurrence of species on croplands. The data set from De Baan et al. (2013b) was used to complement our selection. Ultimately, data on species richness on croplands were collected from 155 publications. In many of these publications, data for multiple cropland types, locations, and/or species groups were reported, which led to a total of 1,053 data points. Information on crop type, data type (i.e., total species richness, richness per field/sample, and diversity indices), taxonomic group studied, sampling method, sampling effort (number of fields, number of samples per field, number of sampling days), sampling period (year, month), location (country, region, coordinates), and farm management (regime years, tillage, crop rotation, pesticide use) were listed, if reported. When coordinates were not reported, Google Earth was used to identify approximate coordinates. The location of each study site was spatially mapped on a global ecoregion and biome map (Olsen et al. 2001) using ArcGIS 9.2.

In order to assess the impact of agricultural land occupation, our method required species richness data from the natural reference situation of each cropland location for the same taxonomic group. Because of limited reference data availability, we combined land use data and reference data from different publications, which was a new approach in this field of work. In our study, we assumed that a reference resembled the typical natural vegetation that would have been if transformation to cropland had not occurred. Since each ecoregion reflects a specific distribution of natural flora and fauna (Olsen et al. 2001), such typical natural vegetation is assumed to be similar throughout an ecoregion. Hence, we paired cropland and reference data from different publications if the studies were located in the same ecoregion. We distinguished three quality levels for the reference data collection. The preferred way to collect reference data was from the same publication (and therefore region) as the cropland data. The next preferred way was to collect reference data from the same ecoregion through a collection of additional publications. If no reference data were found using the above listed options, we selected reference data from the same biome using the set of already selected publications.

Using Eq. 1, CFs were calculated for each pair of cropland and reference situation. Schmidt (2008) states that there are two options when comparing species richness: (1) use a sample size where the species-area curve has reached a clear asymptote or (2) use a standardized area for all land use types (see also Gotelli and Colwell 2001). Here, CFs were derived using reference data obtained with the same sampling technique (e.g., visual survey, quadrat sampling, pitfall traps), the

same sampling area, and the same sampling effort (e.g., number of sampling points, sampling duration) as those of the cropland data. Whether the sampling area and effort were similar between studies, and thus comparable, was decided on a case-by-case basis. Reference data were considered sufficiently similar in case the sampling area and sampling effort of the reference site were within 1 order of magnitude compared to those of the agricultural site. An important assumption that was made was that each of the studies correctly and sufficiently assessed the number of species in the agricultural field or natural area.

When data were available to calculate multiple CFs within one study (e.g., when multiple agricultural fields of the same crop were surveyed for the same species), we averaged these factors to come up with one CF per combination of crop type, species group, and ecoregion, per study. Hereafter, the crops were aggregated into four crop groups, while the taxonomic groups were divided into four broader species groups. The first crop group was oil palm, which was chosen to be a group on its own because of its distinct growth characteristics and because it is a permanent crop rather than an annual crop. The second group includes soybean, sugar beet, potato, and cassava which, based on their low density and height, we expect to provide little shelter to flora and fauna. The remainder of crops were grouped based on taxonomy, distinguishing Pooideae (small cereals: wheat, barley, and rye) and Panicoideae (tall grasses: corn, switchgrass, miscanthus, and sugarcane). Insufficient data were available on sunflower and rapeseed croplands to include these in our assessment. The different species were aggregated based on taxonomy into mammals, birds, arthropods, and vascular plants. Additionally, the data points were spatially reclassified to the biome level, which serves as a proxy of the original vegetation. Of the total 14 terrestrial biomes in the world (Olsen et al. 2001), we had data for 6. A more detailed description of the process of data collection and CF calculation can be found in the Electronic supplementary material (ESM).

2.3 Statistical analysis

Differences between CFs were assessed through two ways of data analysis. In the first approach, median CFs were derived per combination of crop group, species group, and biome (hereafter, *combination-based grouping*). This yielded CFs for specific cases of land occupation, e.g., the impact of corn cultivation on birds in the temperate broadleaf and mixed forest biome. On the downside, structuring our CFs this way limited the statistical analysis as some combinations of categories had to be excluded due to a lack of data. Also, the number of studies available per combination was in some cases rather low, which could influence the outcome of the statistical analysis. In the second approach, median CFs were derived for each crop type, each of the species groups, and each of the biomes, using all available data in that particular category (hereafter, *single-category grouping*). These CFs show the overall median impact (1) of a specific crop on all species groups in all biomes, (2) of all crops on a specific species group in all biomes, and (3) of all crops on all species groups in a specific biome. While this allows for a more straightforward comparison of different crop groups, different species groups, and different biomes, the outcomes of the statistical analysis may be influenced by a skewed distribution of data throughout categories. For example, if the impact of corn cultivation is found to be larger than the impact of soybean cultivation, then this may be due to the fact that more data on vulnerable species are available in corn than in soybean fields. Because of this, conclusions were drawn based on corresponding findings between both grouping approaches.

We tested for statistical differences between combinations of crop groups, species groups, and biomes if the median CFs were derived with data from at least three studies. First, the one-sample Wilcoxon signed-rank test (Wilcoxon 1945) was used to test if the median CFs were significantly different from zero ($p<0.05$). Statistically, a CF that is different from zero implies that the land occupation has a (negative *or* positive) effect on species richness, based on the available data. Second, the median CFs of the different crop groups, species groups, and biomes were compared using the Kruskal-Wallis one-way ANOVA (Kruskal and Wallis 1952). When the Kruskal-Wallis test found significant differences between groups ($p<0.05$), Mann-Whitney *U* tests (Mann and Whitney 1947) were performed pairwise to identify which groups were statistically different. The same method was used previously by De Baan et al. (2013b), except that we adjusted the significance level using the Bonferroni correction for multiple comparisons in order to reduce the probability that nonexistent differences were found. All statistical tests were done in SPSS 19.0 (SPSS Inc., Chicago, IL, USA).

3 Results

Most data on species richness were available for wheat and corn produced in European and North American ecoregions and for oil palm in Southeast Asia (Fig. 1). A total of 309 CFs were calculated, which resulted in 152 CFs on a per-study basis (Table I in the ESM): 51 using only cropland and reference data from the same study, an additional 70 when including ecoregion-based reference data, and 30 more when also including biome-based reference data. These were all calculated based on numbers of species reported, as data availability of other diversity indices was limited. No statistical differences were found between the CFs derived with the three different sources of reference data (Kruskal-Wallis: $p=0.843$). Medians and standard errors (SEs) for the separate groups were similar (ranging 0.43–0.49±0.07–0.10), and

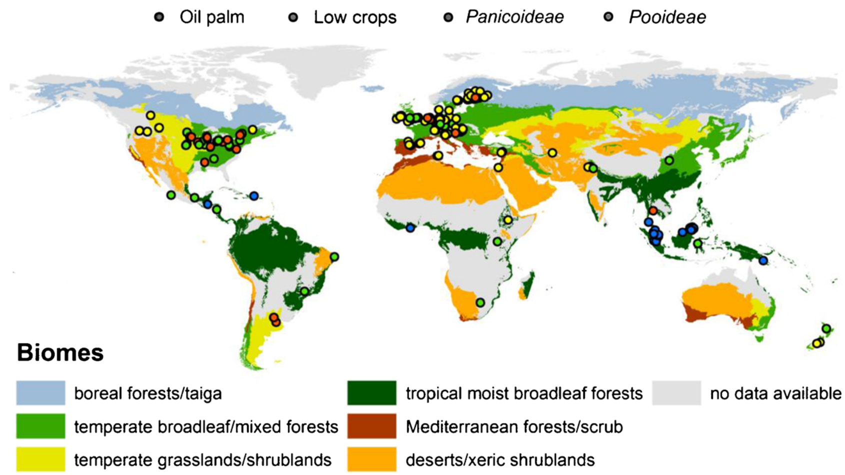

Fig. 1 Locations of cropland data points. Biome map adapted from Olsen et al. (2001). Low crops: cassava, potato, soybean, sugar beet; Panicoideae: corn, miscanthus, sugarcane, switchgrass; Pooideae: barley, rye, wheat

means ranged from 0.24 to 0.38. We therefore consider generating more CFs by adding ecoregion- and biome-based references a valid approach to improve data availability in the present study. Combination-based grouping of the 152 CFs (see Section 2.3) based on the four crop categories and four species categories, and excluding any CFs derived with data from less than three different studies, resulted in a total of 17 median CFs for specific combinations of crops and species

Table 1 Median CFs derived with data from three or more studies, grouped per combination of species group, crop group, and biome. The full list of original CFs can be found in the ESM

Biome[a]	Species group[b]	Crop group[c]	Median±SE	*n* studies	*p* values Wilcoxon[d]
Boreal forests/taiga	Arthropods	Pooideae	−0.44±0.34	5	0.35
		Low crops	−0.55±0.07	4	0.07
	Vascular plants	Pooideae	0.15±0.17	4	0.47
Temperate broadleaf and mixed forests	Birds	Panicoideae	0.62±0.10	3	*0.04*
	Arthropods	Panicoideae	0.35±0.11	14	*0.02*
		Pooideae	−0.03±0.13	16	0.26
		Low crops	0.43±0.17	5	0.08
	Vascular plants	Panicoideae	0.77±0.08	8	*0.01*
		Pooideae	0.72±0.11	11	*<0.01*
		Low crops	0.86±0.05	3	0.11
Temperate grasslands and shrublands	Birds	Panicoideae	0.42±0.10	5	0.11
	Arthropods	Pooideae	0.23±0.18	4	0.29
Mediterranean forests and scrub	Arthropods	Pooideae	0.68±0.10	5	*0.04*
(Sub)tropical moist broadleaf forests	Mammals	Oil palm	0.88±0.16	3	0.11
	Birds	Oil palm	0.72±0.04	7	*0.02*
	Arthropods	Panicoideae	0.51±0.19	3	0.11
		Oil palm	0.53±0.13	16	*0.01*

SE standard error of the mean

[a] All biomes: boreal forests/taiga; deserts and xeric shrublands; Mediterranean forests and scrub; temperate broadleaf and mixed forests; temperate grasslands, savannas, and shrublands; (sub)tropical moist broadleaf forests; (sub)tropical dry coniferous forests; (sub)tropical grasslands, savannas, and shrublands. No data was available for the following biomes: (sub)tropical coniferous forests, temperate coniferous forests, flooded grasslands and savannas, montane grasslands and shrublands, tundra, mangroves

[b] Arthropods: ants, bees, beetles, butterflies, moths, spiders, termites, other arthropods. Mammals: terrestrial mammals and bats. Birds: all. Vascular plants: all

[c] Pooideae: barley, rye, wheat. Panicoideae: corn, miscanthus, sugarcane, switchgrass. Low crops: cassava, potato, soybean, sugar beet. Annual crops: all except oil palm. Permanent crops: oil palm

[d] The one-sample Wilcoxon signed-rank test was used to statistically test whether median CFs were different from zero. This gives an indication whether the agricultural land occupation has any (negative or positive) impact on species richness. Differences were considered significant at $p<0.05$ (shown in italics)

Table 2 Median CFs derived with data from three or more studies, grouped per single category of crop group, species group, and biome. For each category tested, all available data was combined. The full list of original CFs can be found in the ESM

Species group/crop group/biome[a]	Median±SE	*n* studies	*p* values Wilcoxon[a]
Mammals	0.29±0.12	8	*0.02*
Birds	0.62±0.05	25	*<0.01*
Arthropods	0.20±0.09	79	*0.02*
Vascular plants	0.76±0.05	35	*<0.01*
Oil palm	0.62±0.08	27	*<0.01*
Low crops	0.58±0.12	22	*<0.01*
Pooideae	0.20±0.07	57	*0.01*
Panicoideae	0.51±0.13	45	*<0.01*
Boreal forests/taiga	−0.44±0.15	14	0.06
Deserts and xeric shrublands	0.11±0.24	3	0.29
Mediterranean forests and scrub	0.68±0.08	7	*0.02*
Temperate broadleaf and mixed forest	0.40±0.06	72	*<0.01*
Temperate grasslands and shrublands	0.45±0.36	15	*0.02*
(Sub)tropical moist broadleaf forests	0.70±0.08	38	*<0.01*
Annual crops	0.42±0.06	125	*<0.01*
Permanent crops	0.62±0.08	27	*<0.01*
Conventionally managed crops	0.42±0.10	32	*0.01*
Low-input managed crops	0.05±0.11	19	0.67

SE standard error of the mean

[a] See Table 1 for explanation of species group, crop group, biome, and statistical test

within 5 out of 14 biomes (Table 1). Single-category grouping allowed for calculation of median CFs for each of the four species groups, the four crop groups, and a total of four biomes (Table 2).

3.1 Species group

Pairwise comparison of the combination-based grouped data showed that the median CF for vascular plants was larger than the one for arthropods in Pooideae croplands within the temperate broadleaf and mixed forest biome. No other significant differences between species groups were found. After single-category grouping, arthropods were found to have lower median CFs than birds and vascular plants. Notably, nearly one third of the CFs for arthropods were negative, while for mammals, birds, and plants, this was less than 9 % (Fig. I in the ESM). Only 7 out of 17 median CFs significantly differed from zero after combination-based grouping (Table 1), but this can be caused by the small number of studies available for some of these combinations. In the case of the single-category grouped data, the median CFs for all species groups were significantly larger than zero (Table 2).

3.2 Crop type

The median CF of Panicoideae on arthropods in the temperate broadleaf and mixed forest biome was found to be larger than that of Pooideae croplands in the combination-based grouped data, while no significant difference was found between these crop groups after single-category grouping. For the single-category grouped data, the median CF for oil palm was larger than that of Pooideae croplands. One third of the CFs for Pooideae croplands was negative, while oil palm had most CFs in the range 0.5–1.0 (Fig. II in the ESM). All single-category grouped data median CFs were significantly larger than zero (Table 2). These results were derived without accounting for differences in farm management strategy between different croplands because such information was reported sporadically. However, we were able to analyze the impact of farm management in a subset of our data. When grouping all available CFs based on management strategy, a total of 19 CFs were based on low-input farming (including, e.g., organic farming, ecological farming, minimal interference farming, and eco-friendly farm management), and 32 CFs were based on conventional farming (including high-interference farming and high-yield farm management). The data on conventional farms mostly included studies that tested for the impact of herbicides on weeds and invertebrates (indirectly) and a limited number of studies that tested for the impact of insecticide use, either in itself or in combination with herbicides and/or fungicides. The median CF for conventional farming was found to be larger than that for low-input farming (Table 2). Additionally, a paired Wilcoxon signed-rank test was performed to compare a subset of the conventional and low-input farming data, where each pair of data points originated from the same study, thus ruling out the influence of different sampling techniques in different studies.

This test also showed that the number of species in low-input farms was larger than that in conventional farms.

3.3 Region

Data comparison after combination-based grouping showed that the median CF for the Mediterranean forest and scrub biome was larger than that for the temperate broadleaf and mixed forest biome when comparing arthropod richness in Pooideae croplands. Otherwise, no significant differences were found between the median CFs of the biomes. Analysis of the single-category grouped data showed that the median CF for the boreal forests/taiga biome was smaller than those for the biomes Mediterranean forests and scrub, temperate broadleaf and mixed forests, temperate grasslands and shrublands, and (sub)tropical moist broadleaf forests. Also, the CFs for the boreal forests/taiga and deserts/xeric shrubland biomes were not significantly larger than zero, in contrast to the CFs for the temperate and tropical biomes (Table 2). Otherwise, the CFs among different biomes showed no clear differences (Fig. III in the ESM).

4 Discussion

In the present study, we used a data-driven approach to quantify the impact on species richness of multiple species groups when replacing natural vegetation with a variety of agricultural crop types in different regions of the world. We analyzed the results and provide information on the limitations and uncertainties of the empirical approach below.

4.1 Species group

We found that the impact of agricultural land occupation on vascular plants and, possibly, birds is larger than the impact on arthropods. As part of common agricultural practice, most plants are actively removed from croplands, either mechanically or by application of herbicides, to minimize competition for resources and achieve the best possible crop yield. The loss of various plant species is thus a direct effect of agricultural practice. Most native fauna depend on this original vegetation for food, shelter, and breeding or nesting (Wilson et al. 2005) and will be impacted accordingly. However, some of the original animal species may maintain themselves in the new croplands ("shared species"; e.g., Danielsen et al. 2008; Estrada and Coates-Estrada 2005; Ottonetti et al. 2010; Ouchtati et al. 2012). Other species prefer farmland habitats and are absent or less abundant in the natural situation (e.g., Gaines and Gratton 2010; Ottonetti et al. 2010). These farmland species mostly include invertebrates like beetles, spiders, and butterflies, as well as some bird species (Wilson et al. 2005). Arrival of such species increases the field species richness, thereby lowering the CF and masking the loss of native invertebrates. Most studies that reported increasing species of arthropods were on ground beetles (Carabidae), which was also the species group for which most data were available. A handful of studies reported larger numbers of rove beetles, ground-dwelling spiders, springtails, and moths. Most ground beetles feed on invertebrate prey and could be beneficial to farmers. The same is applicable for rove beetles and spiders. Moths (or their caterpillars), on the other hand, can be a major agricultural pest. However, it was outside the scope of this study to check for the ecological value of either the native species lost or the species newly arrived upon agricultural land occupation.

De Baan et al. (2013b) reported median CFs of 0.56 and 0.65 for arthropods in permanent and annual crops, respectively. For permanent crops, i.e., oil palm, we found a similar value (0.53 ± 0.13), but our median CF for arthropods in annual crops differs considerably (0.11 ± 0.10). For vascular plants in annual croplands, De Baan et al. (2013b) found a median CF of 0.42 versus 0.76 in the present study. For birds, similar results were found: 0.62 and 0.53 by De Baan et al. (2013b) for permanent and annual crops, respectively, versus 0.72 ± 0.04 and 0.58 ± 0.06 in the present study. Given that the methods of the present study and that of De Baan et al. (2013b) are similar, the differences between some of our outcomes may be attributed to an extended data set from our side.

The present study covered a wide spectrum of taxonomic (sub)groups, but the majority of published data were on plants (27.9 %), ground beetles (12.5 %), ground-dwelling spiders (10.3 %), and birds (5.9 %). Birds are typically well represented in literature studies because they are easily surveyed, they are taxonomically well known, and general support for conservation of birds is high (Larsen et al. 2012; Rodrigues and Brooks 2007; Vandewalle et al. 2010). The high availability of plant, beetle, and spider data can be explained by the interests of the agricultural sector. Plants (or "weeds") are actively removed from agricultural fields because they compete for resources with the crops of interest. Many of the collected publications were dedicated to the testing of different herbicides or mechanical efforts on the removal of plants (e.g., Mulugeta et al. 2001; De Snoo 1997; Ulber et al. 2009). The opposite holds for ground beetles and spiders, some of which predate on pest species (Booij and Noorlander 1992; Kromp 1999; Lang et al. 1999; Riechert and Bishop 1990) and thus are beneficial to agriculture. Multiple collected publications were dedicated to studying the impact of farm management on ground beetle or spider communities (e.g., Basedow 1998; Booij and Noorlander 1992; Boutin et al. 2009; Schmidt et al. 2005). Also, insects are numerous and diverse, can be sampled relatively easily, and are responsive to environmental change (Vandewalle et al. 2010).

4.2 Crop type

The results showed that the impact of oil palm cultivation is larger than the impact of Pooideae cultivation. However, since oil palm is only grown in (sub)tropical regions and our data on Pooideae croplands are almost exclusively from temperate, boreal, and Mediterranean regions, we cannot rule out that the difference between these crops is actually caused by a difference between these biomes. The same applies to the difference that was found between the median CFs of annual and permanent crops since oil palm is the only permanent crop included. We would have expected a larger impact of annual crops since the annual harvest and subsequent tillage of the soil frequently disturb the farmland habitat, while plantations of permanent crops like oil palm can be a constant, relatively undisturbed habitat for longer periods. The larger impact of the permanent crop could be caused by the data origin. All oil palm data originated from (sub)tropical regions that have a relatively large species richness in the natural habitat, while the data on annual crops mostly originated from the temperate and boreal regions. Therefore, we do not recommend the use of our oil palm-based CF for permanent crops in the impact assessment of permanent crops typically grown in more temperate regions. Likewise, the CF for annual crops should only be used for the temperate and boreal regions. Our study did not confirm the findings of earlier smaller scale studies that found a larger species richness in cellulosic crops (miscanthus and switchgrass) than in corn and wheat (Bellamy et al. 2009; Gardiner et al. 2010; Ward and Ward 2001). More data are needed to derive more concrete conclusions.

Regarding farm management, we found that conventional farming, which often includes pesticide use and tillage, has a larger impact on biodiversity than low-input farming, producing median CFs of 0.42±0.10 and 0.05±0.11, respectively. This is in line with earlier findings by Köllner and Scholz (2008), who used central European data on plant species richness to calculate CFs for ecosystem damage and reported 0.63 for high-intensity agriculture and −0.06 for low intensity agriculture. De Schryver et al. (2010) came to the same conclusion, although their median CFs were larger (0.36 for organic and 0.79 for intensive arable land). Likewise, Schmidt (2008) found that the impact of intensive cultivation of cereals and annual crops was larger than the impact of extensive cultivation, although our numbers are not readily comparable. Finally, Müller et al. (2013) reported CFs of 0.60 and 0.15 for conventional and organic cultivation of fodder crops in temperate regions, respectively, and CFs of 0.81 and 0.42 in tropical regions, when assessing its impact on plant species richness.

The focus of the present study was on a limited number of crops that are relevant as food crops and/or biofuel feedstocks. It may be possible to estimate the impact of some other crops based on its shared growth characteristics with the different crop groups in this study. For example, the impact of cultivation of grasses such as sorghum and reed canarygrass is expected to be similar to the impact of the included Panicoideae, and cultivation of numerous vegetables (e.g., alliums, legumes, salad crops) is expected to have an impact similar to our low crops group. In such cases, the CFs derived in the present study could be used. In other cases, correlation based on growth characteristics or taxonomy may not apply because of differences in farming technique. For example, the impact of wet rice cultivation is expected to be very different from that of larger Pooideae grown on dry grounds. Low data availability hinders the impact assessment of many additional crops.

4.3 Region

The impact of agricultural land occupation was found to be lower in boreal forests/taiga than in most other biomes. Only deserts/xeric shrublands showed no statistically significant difference from boreal forests/taiga. Boreal forests/taiga and deserts/xeric shrublands are also the two biomes for which no statistically significant effect of land occupation was found (i.e., their mean CFs were not statistically different from zero). Like in any region, land occupation in these harsh environments will cause a decrease of sensitive species, but agricultural practice like irrigation and a potential increase of food resources will also attract various opportunistic species (Cook and Faeth 2006; Khoury and Al-Shamlih 2006). When using species richness as an indicator, this can result in lower CFs. In the four other biomes tested (i.e., temperate broadleaf and mixed forests, temperate grasslands and shrublands, Mediterranean forests and scrub, (sub)tropical moist broadleaf forests), agricultural land occupation was found to have a negative effect overall, with the exception of arthropods in temperate grasslands and shrublands. Likewise, De Baan et al. (2013b) found a small positive effect of land occupation in the deserts and xeric shrublands biome and a negative effect in the temperate, (sub)tropical, and Mediterranean biomes. Comparing our results for annual crops with those from De Baan et al. (2013b), we found a lower impact in the temperate broadleaf and mixed forest biome (0.40±0.06 versus 0.76) and a larger impact in the (sub)tropical moist broadleaf forest biome (0.83 ±0.11 versus 0.54), likely because different selections of crops were included in our data sets. No other studies provided impacts of annual crops on a biome basis.

4.4 Uncertainties and limitations

We chose relative species richness as an indicator, because it captures biodiversity at the community level (see, e.g., Curran et al. 2011) and because it is reported frequently. However, relative species richness only provides information on a small aspect of biodiversity and several limitations can be identified.

A major drawback of using species richness as an indicator of biodiversity is that it does not take species abundance into account (Larsen et al. 2012). Using this indicator, there is no distinction between a single individual of one species (perhaps an accidental encounter) and a large healthy population of another species; both count as one species. This limitation may be covered by using other biodiversity indicators such as the Shannon index or mean species abundance of original species (Alkemade et al. 2009; Hanafiah et al. 2012), but these indicators require data that are rarely reported on a global scale. The choice of indicators that are most suitable and meaningful is subject to debate (Geyer et al. 2010). De Baan et al. (2013b) were able to compare impacts across five different biodiversity indicators (relative species richness, Fisher's α, Shannon's H, Sørensen's S_S, and mean species abundance) for one biome and found considerable variation therein but concluded that relative species richness is the most suitable indicator in view of current data availability. Likewise, Vogtländer et al. (2004) compared the species richness indicator with an ecosystem indicator that is based on ecosystem richness, diversity, and rarity. They also found that species richness is an accurate proxy of biodiversity in most cases. The species richness data we collected can be considered indicative of differences between crops, species, and regions. It should be noted, however, that native and farmland species were treated equally in the calculations of our CFs, as information on the species' origin was not provided in the majority of cases. However, arriving farmland species may be considered of less (ecological or societal) value than native species, or they may even be pests, so such new arrivals should not be considered a compensation for a loss of native species. Indicators that compare exclusively the richness or abundance of native species between a reference and land use situation, e.g., Sørensen's S_S (Sørensen 1948) and mean species abundance (Alkemade et al. 2009), are more sensitive to land use impacts than relative species richness (De Baan et al. 2013b). Consequently, our results could underestimate the impact of agricultural land use on the *native* biodiversity.

Even though species richness is studied and reported relatively often compared to other biodiversity indicators, for many combinations of crop types, species groups, and biomes, too little data were available to derive CFs or perform sound statistical analysis. Low data availability has been a limiting factor throughout the present work. Still, we were able to increase our data set size by nearly two-thirds by combining land use and reference data from different sources. We here considered this approach valid as we found no significant differences between the outcomes of all three data quality levels. However, it is important to emphasize that pairing data from two different publications should not be preferred over using data from the same study because the uncertainty of the outcomes increases when the two publications use (slightly) different sampling methods or effort.

The incapability to fully cover biodiversity is an important limitation in land use impact assessments. Often, vascular plant species are used as proxy for the impact on total biodiversity (Köllner 2000; Lindeijer 2000b; Schmidt 2008). In the present study, vascular plants were shown to be the most sensitive group. Therefore, concentrating on vascular plants as an indicator of biodiversity may entail overestimation of the impact of agricultural land occupation on overall species richness. This is in line with the lack of success to use single taxonomic groups as an indicator for overall biodiversity (e.g., Larsen et al. 2012; McGeoch 1998; Michelsen 2008; Prendergast 2006; Schulze et al. 2004). As long as there is no consensus on which (combinations of) species groups to use as indicators, biodiversity impact assessments should be performed including a diversity of taxonomic groups in order to cover total biodiversity as comprehensive as possible. More effort should be taken to study the impact of land occupation on underexposed groups like amphibians and reptiles.

Some aspects of agricultural land occupation that potentially influence its impact on biodiversity were excluded in the present study. This includes specific choices in farm management (pesticide use, fertilization, tillage, crop rotation, intercropping), farm age, and the layout of the surrounding landscape (proximity to natural vegetation, habitat heterogeneity). The impact of most of these factors on farm biodiversity has been shown in various field experiments (e.g., Benton et al. 2003; Hole et al. 2005; McLaughlin and Mineau 1995; Meek et al. 2002) but remains neglected in the current land use impact assessment methodology. While farm management information was found to be rarely reported in detail, sufficient data should be available locally (e.g., for management of cereal fields in the UK). Such local data can help to quantify the influence of farm management types in regional land use assessments. Finally, it is important to note that all the results and analyses are valid only within the framework of the assumptions made and the use of aggregated data. Increasing the granularity in the data may change some of the findings. It is recommended that more data are generated and collected in order to improve the statistical analysis and increase the accuracy of results.

5 Conclusions

The main utility of this work was to provide insight into the factors that influence the impact of land occupation on biodiversity. We calculated CFs and performed statistical analysis to assess the effect of land occupation on multiple species groups in different regions of the world.

Different species groups were found to respond in a different way to agricultural land occupation, so focusing only on "indicator species" in life cycle impact assessments is undesirable. Whenever possible, species of multiple groups should be included. Vascular plants (most sensitive) and arthropods

(least sensitive) should at least be well represented. As significant differences were found between some biomes, the impact of land occupation should not be quantified at a global level but rather at a biome or ecoregion level. Farm management strategy should be included as an additional factor whenever data is available, as it was found to be of significant influence. However, differentiation between specific crops seems redundant considering that we found no significantly different impacts between crop groups based on density and height using the collected data. Our data also suggests that a distinction between annual and permanent crops may be more useful. The CFs derived in the present study are an important addition to the already available set of CFs to further improve impact assessments of agricultural land use. Additional and more granular data are needed to improve accuracy of the assessments.

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